

6 Chemical Transport and Fate

6.1 Introduction

This section summarizes the physical and chemical properties of the chemicals of concern (COCs) that influence their transport and fate in the Lower Fox River and Green Bay. Transport and fate processes affect how these compounds behave in the natural environment and move from source areas to potential human and environmental receptors. The discussion addresses PCBs, dioxin, dieldrin, DDT, arsenic, lead, and mercury, which were identified in the SLRA as chemicals of potential concern. In addition, the models to be used for the evaluation of PCB fate/transport and bioaccumulation are introduced and summarized. However, the results are not presented herein but can be found within the documentation report for each model.

A number of important physical and chemical processes affect contaminant fate and transport:

- Wind and stream flow/water movement
- Chemical-specific factors affecting partitioning between solid and dissolved phases
- Interactions between the dissolved and particulate phase of each compound (or compound group) within sediment, water column or biota
- Chemical partitioning or transformation in sediment, water, or biota

Once discharged into the environment, all of the chemicals discussed herein partition to sediment particles to some degree. Chemicals adsorbed onto the sediments are predominantly transported within the river system by physical processes. Important chemical and biological processes which facilitate uptake within the food chain include partitioning coefficients, metabolic processes, and species-specific bioaccumulation or bioconcentration factors. These processes are discussed in greater detail below.

6.2 Transport and Fate Processes

6.2.1 Chemical Transport Interactions

Chemical transport occurs through a variety of processes, including the following:

- Dissolution in the water column
- Volatilization into the atmosphere
- Adsorption to sediment (which may be deposited or suspended in the water column) and/or
- Incorporation into the food chain

In general, the chemicals of potential concern (COPC) tend to sorb to sediment particles, are resistant to biodegradation, volatilize slowly and bioaccumulate in aquatic organisms. Some chemical-specific measures, which are generally interrelated, that affect these tendencies include the following:

- Water solubility
- Organic carbon partitioning coefficient (K_{oc})
- Octanol-water partitioning coefficient (K_{ow})
- Vapor pressure
- Henrys Law constant (H_c) vapor: water partitioning coefficient
- Biodegradation rate
- Bioaccumulation factor

The water solubility of a chemical partly determines the extent to which a substance can partition between sediments and pore water/surface water. Because water is a polar solvent, polar covalent and ionic compounds are more likely to dissolve than non-polar compounds. Dissolution of non-polar organic chemicals are further controlled by their affinity for organic carbon phases in sediments or water. Both the K_{oc} and K_{ow} partitioning coefficients may be used to predict the degree of chemical sorption to organics in soil, sediment and particulate matter. The higher the K_{ow} , the greater affinity for partitioning to organic carbon. Vapor pressure and the Henrys Law constant are an indication of how readily a compound will volatilize from water into the atmosphere. The biodegradation rate, when known, provides the rate at which microbial processes may be expected to break down a chemical. Although the bioaccumulation factor is not a specific chemical property, it is a function of K_{oc} and can enable estimates of the degree to which a given chemical may be expected to be incorporated into tissues of aquatic organisms.

These are usually the most important factors effecting the overall fate of a chemical in the environment and they can be used to predict the mechanisms by which each contaminant (or group of contaminants) will move through or transform in the

environment. Typical values for some of these chemical factors are included on Table 6-1 to enable general comparison with the other chemicals in the system.

6.2.2Physical Transport

Flowing water is the primary transport mechanisms for movement of contaminated sediment in the Lower Fox River downstream to Green Bay. Additionally, bay currents move sediments from southern Green Bay along the east shore, as discussed in Sections 3 and 5. Sediment transport is the primary mechanism for chemical movement in the Lower Fox River and Green Bay.

Surface water transport mechanisms depend on the type of water body present. In the Lower Fox River, the water velocity and sediment particle characteristics are the two main factors which influence the physical movement of sediment and the chemicals adsorbed onto their surfaces. The stream flow characteristics and physical movement of sediment particles as TSS in the Lower Fox River and Green Bay were discussed in detail in Section 3.

Chemicals sorbed to sediments and organic matter may be transported in suspension or as bed load by river currents. Fine-grained material, such as silts and clays, will generally be entrained in the water column and migrate downstream as suspended solids. As water velocities increase due to storm events or seasonal runoff, coarser-grained material (medium to coarse-grained sand or larger particles) will become suspended and/or move along the river bottom as bed load. Chemicals may accumulate as deposits as river velocities decrease. After deposition, bottom sediments are subject to resuspension.

In the case of larger water bodies, such as lakes (e.g., LLBdM and Green Bay), chemical/sediment transport occurs through wind and water driven currents as well as wave action. In general, currents are relatively slow and transport only fine-grained material. Large waves along near shore areas of Green Bay and Lake Michigan are capable of moving boulder size particles along the shoreline.

If a chemical dissolves in surface water, its chemical transport properties will be identical to those of water. Compounds present as an immiscible liquid phase will either sink or float on water depending on the compound's specific gravity.

Nonaqueous-phase liquids with a specific gravity of less than one will tend to remain close to, or float on the surface and may become susceptible to attenuation by volatilization and photolysis. Immiscible liquids more dense than water will move along the river bottom and/or become absorbed onto sediment particles.

Substances dissolved in surface waters can also partition out of the dissolved phase to a liquid phase or adsorb onto particles suspended in the water or onto bottom sediment. The latter process transfers the substances from the water to the sediment matrix. Conversely, chemicals may desorb from sediment back into the water.

Dispersion is a rapid process because of turbulent eddying (advection) and diffusion along concentration gradients. The amount of dilution can be approximated by comparing rates of chemical introduction to river flow discharge rates. In stagnant water bodies, such as marshes, advective forces are less important, and primary attenuation may be through diffusion.

6.2.3 Biological Interactions

Other important processes that affect long-term chemical persistence include bioturbation of sediments and bioaccumulation. Sediment bioturbation will generally improve degradation rates of organic compounds through oxygenation of surface sediments. Although bioturbation can have an effect on the anaerobic dechlorination of PCBs, it has little impact on the degradation rates of inorganic metals.

Bioaccumulation occurs in an organism when the uptake rate exceeds the organisms ability to remove the chemical through metabolic functions, dilution, or excretion, so that the excess chemical is stored in the body of the organism. One result of bioaccumulation may be biomagnification of the chemical up the food chain. Biomagnification occurs at the upper end of the food chain when the chemicals are passed from one organism to another through consumption (e.g., phytoplankton contain low levels of PCBs which are passed to the fish and ultimately to piscivorous birds or humans).

Benthic infauna occur in the upper strata of sediment in the Lower Fox River and Green Bay. Sediment is mixed by these organisms throughout their life cycles. The depth of sediment that is susceptible to mixing by various infaunal organisms varies with the sediment grain size, density, sediment chemistry, bottom current velocity, and type of habitat available. Benthic insect larvae, ingest bulk sediment and strip detritus from the surface of the particles. PCBs (and other chlorinated compounds) partitioned to sediments may enter into the food web principally from uptake of sediment solids (Capel and Eisenrich, 1990). Various oligochaetes (worms) and chironomid larvae (insects) were observed to depths up to 2 feet in the Lower Fox River deposits (GAS/SAIC, 1996), suggesting that bioturbation in the system may occur in the upper 2 feet of the river and bay sediments.

6.3 Compounds of Potential Concern

The following summaries were largely derived from the Agency for Toxic Substances and Disease Registry (ATSDR) Toxicological Profiles for each compound or group of compounds discussed.

6.3.1 Organic Constituents

6.3.1.1 PCBs

PCBs exhibit low water solubility, are moderately volatile, strongly adsorb to organics, and preferentially partition to soil and sediment. The major fate process for PCBs in water is adsorption to sediment or other organic matter. Consequently, PCB concentrations in sediment and suspended matter are generally higher than in the associated water column (ATSDR, 1997a). The more highly chlorinated Aroclors sorb more strongly than the less chlorinated Aroclors, reflecting their differences in water solubilities and octanol-water partition coefficients. Adsorption and subsequent sedimentation may immobilize PCBs for relatively long periods of time in aquatic systems. However, redissolution into the water column may occur. PCBs contained in layers nearest the sediment surface may be slowly released over a long period of time. PCBs present in the lower layers of sedimentary deposits may be effectively sequestered from environmental distribution (ATSDR, 1997a).

The estimated Henry's law constants for individual Aroclors indicate that volatilization may be a significant environmental transport process for PCBs dissolved in natural water. However, adsorption to sediment significantly decreases the volatilization rate of highly chlorinated Aroclors from the aquatic phase. The redissolution rate of PCBs from sediment to water is greater in the summer than in the winter because of more rapid volatilization from water at higher temperatures.

The ability of PCBs to bioaccumulate has been related to corresponding octanol-water partition coefficients (K_{ow}). Compounds with high K_{ow} values more readily bind to sediments (particularly sediments with elevated organic carbon) and are more readily bioaccumulated by organisms. Experimentally determined bioconcentration factors (BCFs) in freshwater aquatic animals range from 600 to 274,000 (ATSDR, 1997a). The BCFs in aquatic animals may depend on the water depth in which they predominantly feed. Certain benthic organisms accumulate PCBs from water at the water/sediment interface and via intake of phytoplankton and zooplankton which contain higher levels of PCBs than the surrounding water (ATSDR, 1997a). In addition, the bioconcentration of PCBs in bottom-feeding species is also expected to be high because the PCB concentrations in sediment are several orders of magnitude higher than those in water.

PCBs also biomagnify within the food chain, as indicated by the PCB concentrations in higher trophic levels of aquatic organisms and in several species of piscivorous birds and seals (ATSDR, 1997a). The biotransfer factors of Aroclor 1254 were estimated to be 0.052 and 0.011 kg/day, “respectively, while the estimated mean BCF value for PCBs in human fat was 128 (ATSDR, 1997a). The BCF value is the PCB concentration in tissue over the PCB concentration in the diet.”

The ability of PCBs to be degraded or transformed in the environment depends on the degree of chlorination of the biphenyl molecule in addition to the isomeric substitution pattern (ATSDR, 1997a). Aroclor 1242 appears to be persistent in this aquatic environment, given that it was the PCB mixture used by the PRPs in the production/recycling of carbonless copy paper and due to the detected concentrations and predominance of this mixture in both river and bay sediments.

Analysis of the movement of PCBs in the Lower Fox River was evaluated in several investigations. Studies indicate that suspended solids are the most important factor in the transport and fate of PCBs (Gailani, 1991; Velleux and Endicott, 1994; WDNR, 1995; and Velleux, *et al.*, 1995). WDNR (1995) found a strong correlation and dependency between total PCB (dissolved and particulate) and suspended solids concentrations in the water column. This PCB-suspended solids correlation was not observed at the upstream boundary where PCB concentrations leaving Lake Winnebago were low, ranging from 1 to 3 ng/L.

6.3.1.2 Dioxins and Furans

Chlorinated dioxins are a group of over 75 different compounds, and chlorinated dibenzofurans comprise over 135 compounds (ATSDR, 1989). These compounds have been found to be very persistent in the environment due to their low solubility in water and affinity for organic matter in sediments. The fate of 2,3,7,8-TCDD is not understood with certainty. According to some studies, surface water sediments are the ultimate environmental sink of airborne particulate 2,3,7,8-TCDD (ATSDR, 1997a).

2,3,7,8-TCDD is expected to be immobile in most soils. A rate of transport of 10 cm in 12 years has been observed with soils from Elgin Air Force Base. Leaching is possible, but very unlikely, in soils with a very low organic carbon content. Bacterial degradation of dioxins and furans is possible but it is a very slow process and is usually limited by the populations of organisms in the native material. However, both volatilization and photolysis will remove 2,3,7,8-TCDD from surface soils. Therefore, the half-life in surface soils may range from 1 to 3 years but for contaminants buried a few inches below the surface the half-life may be 10 to 12 years or more (ATSDR, 1989). In surface and groundwater, degradation of

2,3,7,8-TCDD is similar to that in soils, and volatilization and photolysis are again the two most significant processes (ATSDR, 1989).

Dioxins and furans have been found to highly bioconcentrated in aquatic organisms and wildlife. Experiments with fathead minnows (*Pimephales promelas*) have yielded a BCF of 7,900 to 9,300 on a wet weight basis. Similarly, studies completed on rainbow trout (*Salmo gairdneri*) have been extrapolated to yield a BCF of approximately 39,000 (ATSDR, 1989).

6.3.1.3 DDT

DDT, and its principal metabolites DDD and DDE, are organochlorine compounds that were used as insecticides until 1972, when their use in the United States was banned because of adverse toxicity to wildlife. These compounds may be transported from one medium to another by adsorption, bioaccumulation, dissolution, or volatilization. The major fate process for DDT in water is adsorption to sediment or other organic matter and the primary loss route is the transportation of the particulates to which the compound is bound (ATSDR, 1994). Studies have shown that in soils/sediments, DDT transformations have prolonged persistence. These compounds undergo extensive adsorption to soil particles, especially those with high TOC levels.

Photo-oxidation of DDT is known to occur on soil surfaces; however, it is not known to hydrolyze (ATSDR, 1994). Biodegradation may occur under both aerobic and anaerobic conditions by microorganisms including fungi, algae and mixed microbial populations. Under aerobic conditions DDT slowly converts to DDE whereas under anaerobic conditions it converts to DDD much more rapidly. However, the estimated DDT half-life ranges from 2 to more than 15 years (ATSDR, 1994; Stewart and Chisolm, 1971).

Studies have found that plants, fish, mammals, and birds, as well as phytoplankton and zooplankton in an aquatic environment, bioaccumulate DDT. DDT has a high potential to bioaccumulate and the BCF for rainbow trout has been estimated to be 12,000 while the estimated human BCF is above 1,650, primarily from the consumption of fish (ATSDR, 1994). A study completed in northern Canada found that biota living in the bottom of the sea had much higher levels of total DDT than biota living in the open sea (ATSDR, 1994). This is likely a result of DDT adsorption onto particulates which settled into bottom sediments. Additionally, the ring necked seal apparently biomagnified DDT, indicating that biomagnification is possible in other species as well (ATSDR, 1994). Others have also found DDT biomagnification from soil sediment to mosquito fish, and a study completed in Lake Michigan indicated that DDE biomagnified 28.7 times from

phytoplankton to fish and 21 times from sediments to amphipods (ATSDR, 1994).

In sediments, DDE is the major metabolite formed (Montgomery, 2000). Both DDD and DDE are stable and biologically active, but DDE is non-insecticidal (Montgomery, 2000). DDT has a low solubility and preferentially binds to sediments. If consumed, DDT and metabolites are stored in fat and, as shown above, biomagnify up the food chain.

6.3.1.4 Dieldrin

Dieldrin is both a manufactured pesticide and a breakdown product of the pesticide aldrin. Therefore, the presence of dieldrin in the environment may result from either the application and use of dieldrin or aldrin. The use of both products has been banned in the United States since 1974. Therefore, the presence of dieldrin in river sediments is due to their continued persistence in the environment.

Dieldrin is extremely non-polar, has a low volatility, sorbs readily to sediment organic matter and has a high potential for bioaccumulation with a BCF of 4,670 (ATSDR, 1993a). Dieldrin is persistent in sediments and surface water with a half-life of 3 and 6 years, respectively (Howard, 1991). Direct photolysis of dieldrin can occur creating a half-life of about 2 months (ATSDR, 1993a). Dieldrin degradation is unaffected by aerobic or anaerobic conditions (Montgomery, 2000), but dieldrin can be biotransformed by soil microbes to a substance more toxic to insects. The persistence of dieldrin within soil and sediment is exemplified by a study in soil plots which had been treated with dieldrin for 15 years. The dieldrin concentrations were the same 4 years after treatment stopped (ATSDR, 1993a).

Volatilization is the principle route of loss from soil; however, the process is slow due to the low vapor pressure. Once in the atmosphere, dieldrin can travel great distances. Studies in the Northwest Territories of Canada have found mean concentrations of 0.75 ng/L in arctic snow. There were no known local sources.

Experimental evidence has shown that aldrin converts rapidly to dieldrin, which readily bioaccumulates and biomagnifies (ATSDR, 1993a). Radiolabeled aldrin was added to an ecosystem and was converted to dieldrin. Sampling results indicate that of the radiolabeled aldrin, approximately 96 percent of the total stored in fish, 92 percent of the total found in snails, and about 86 percent of the total found in algae was in the form of dieldrin (ATSDR, 1993a). Measured BCFs are approximately 2,700 in fish and 61,657 in snails. In rainbow trout, a biomagnification factor of 1.0 has been determined on a lipid weight basis and 2.3

for the wet weight. Additionally, biotransfer factors evaluated for beef and cow milk were estimated to be 0.008 and 0.011, respectively (ATSDR, 1993a). These results all indicate the strong affinity of dieldrin for organic matter and the increased likelihood of biomagnification to higher trophic levels of the food chain.

6.3.2 Inorganic Constituents

The primary factor influencing the fate and transport of heavy metals is their speciation and adsorption capacity, which are affected by and change with the geochemistry of the environment. The transport of metals via surface water is affected by adsorption of metals to soil or other organic matter. The degree to which a metal will adsorb depends on the presence of competing ions, water chemistry, and metal speciation, which is, in turn affected by such factors as pH and reduction/oxidation (redox) potential. The interaction among these factors is complex.

In instances where metals are present in solution with other ions, competition for sorption sites on soil particles or on organic material may enhance the mobility of weakly sorbed metals such as cadmium. Adsorption of metals is also strongly influenced by pH, due in part to increased competition between protons (H^+) and metal ions for the same binding sites. Furthermore, pH affects the speciation and solubility of metals through the formation of hydroxide complexes. Speciation of metals is also controlled by the redox potential of the environment, which determines the oxidation state of the metal. The fate and transport of individual metals of concern are discussed below.

6.3.2.1 Mercury

The transport and partitioning of mercury in surface waters and soil is influenced by the particular form of the compound. Volatile forms (e.g., metallic mercury and dimethylmercury) evaporate to the atmosphere, whereas solid forms partition to particulates or are transported in the water column, depending upon their solubility. However, the dominant process controlling the distribution of mercury compounds in the environment is the sorption of non-volatile forms to soil and sediment particulates. The sorption process is related to the organic matter content of the soil or sediment; the pH of the medium apparently does not affect the process. Inorganic mercury sorbed to particulate material forms stable complexes with organic compounds and is not readily desorbed or removed from sediment (ATSDR, 1997b). Consequently, freshwater and marine sediment are repositories for inorganic forms of mercury. Mobilization of sorbed mercury from particulates can occur through chemical or biological reduction to elemental mercury and bioconversion to volatile organic forms (ATSDR, 1997b).

Methylmercury produced from biotransformation processes is soluble and mobile, and quickly enters aquatic food chains. Methylmercury is not only the most biologically available form of mercury, it is also the most toxic. Mercury bioaccumulation and biomagnification have been demonstrated in the aquatic food chain by the elevated levels found in piscivorous fish compared with organisms lower on the food chain. Almost all mercury accumulated is in the methylated form, primarily as a result of the consumption of prey containing methylmercury. Methylmercury accumulates in carnivorous fish to levels of 10,000 to 100,000 times those found in ambient water (ATSDR, 1997b). Bioaccumulation of methylmercury in aquatic food chains is of interest because it is generally the most important source of nonoccupational human exposure to the compound (EPA, 1984). Mercury methylation in ecosystems depends on mercury loadings, microbial activity, nutrient content, pH, redox conditions, suspended sediment load, sedimentation rates, and other variables (ATSDR, 1997b). Conversion of inorganic mercury to methylmercury is favored by low pH and low dissolved organic carbon levels.

6.3.2.2 Lead

The primary sources of lead in the environment are anthropogenic emissions to the atmosphere. A significant fraction of lead carried by river water is in an undissolved form, consisting of colloidal particles or larger undissolved particles of metallic lead, lead carbonate, lead oxide, lead hydroxide, or other lead compounds incorporated in other components of surface particulate matter from runoff. Lead may occur either as sorbed ions or surface coatings on sediment particles, or it may be carried as a part of suspended living or nonliving organic matter in water. Lead in aquatic environments often precipitates out of solution by binding to carbonate or phosphate ions or it can be readily sorbed to either organic or inorganic components in sediments.

Factors affecting the degree of sorption include: the sediment type, pH, organic carbon content, cation exchange capacity, the form of the lead and other constituents in the sediment such as metal oxides, aluminum silicates, and carbonates. Sorption is high in sediments containing clay and lower in sediments containing a higher percentage of sand or sand and loam (ATSDR, 1993c). Lead can bioaccumulate but does not biomagnify.

Plants and animals may bioconcentrate lead. In general, the highest lead concentrations are found in organisms near lead mining, smelting, and refining facilities; storage battery recycling plants; areas affected by high automobile and truck traffic; sewage sludge and spoil disposal areas; sites where dredging has occurred; areas of heavy hunting (from spent shot); and in urban and industrialized areas. Lead is not biomagnified in aquatic or terrestrial food chains.

Older organisms tend to contain the greatest body burdens of lead. In aquatic organisms, lead concentrations are usually highest in benthic organisms and algae, and lowest in upper trophic level predators (e.g., carnivorous fish). High BCFs were determined in studies using oysters (6,600 for *Crassostrea virginica*), freshwater algae (92,000 for *Selenastrum capricornitum*), and rainbow trout (726 for *Oncorhynchus mykiss*), although most median BCF values for aquatic biota are significantly lower: 42 for fish, 536 for oysters, 500 for insects, 725 for algae, and 2,570 for mussels (ATSDR, 1993c). Lead is toxic to all aquatic biota, and organisms higher on the food chain may experience lead poisoning as a result of eating lead-contaminated food.

6.3.2.3 Arsenic

Most naturally occurring arsenic in the environment exists in soil or rock. This material may be transported by wind or water erosion of small particles. Arsenic can also leach from soil or rock into rainfall or snow melt (ATSDR, 1993b). However, because many arsenic compounds tend to adsorb to soil or sediment, leaching usually results in transportation only over short distances (EPA, 1982; Welch, *et al.*, 1988).

Transport and partitioning of arsenic in water depend upon its chemical species, oxidation state, and on interactions with other materials present. Soluble forms may be carried long distances through rivers (ATSDR, 1993b). However, arsenic may be adsorbed from water onto sediment or soil, especially clays, iron oxides, aluminum hydroxides, manganese compounds, and organic material (Callahan, 1979; EPA, 1982; Welch, *et al.*, 1988). Sediment-bound arsenic may be released back into the water by chemical or biological interconversions of arsenic species. In an oxidized environment, arsenic is generally present as arsenate (As^{5+}), an immobilized form that will be ionically bound to soil. However, under reduced conditions, arsenate is transformed to arsenite (As^{3+}), which is water soluble and, therefore, more mobile.

Arsenic present in Lower Fox River sediment may be associated with agricultural chemicals such as pesticide and herbicides and with wood treatment facilities. Arsenic in this form is tightly bound and generally not as bioavailable as soil- or sediment-bound arsenic. Bioconcentration of arsenic occurs in aquatic organisms, primarily in algae and lower invertebrates. BCFs measured in freshwater invertebrates and fish for several arsenic compounds ranged from 0 to 17 (EPA, 1980) and biomagnification in the aquatic food chain does not appear to be significant (ATSDR, 1993b; EPA, 1982).

6.4 Lower Fox River/Green Bay Modeling

Computer models have been employed in the RI/FS/RA to assist in the evaluation PCB fate and transport, historically and into the future. These models also enable the evaluation of various remedial scenarios on future PCB distribution in various environmental media as well as the food web in the Lower Fox River and Green Bay. These models are briefly described below and additional information is included in the documentation report for each specific model.

6.4.1 GBTOXe Model

The enhanced Green Bay toxics model (GBTOXe) was developed by HydroQual to simulate the fate and transport of PCBs in Green Bay for the RI/FS. GBTOXe is an enhanced version of an existing WASP4 based toxics model developed as part of the GBMBS by Bierman, *et al.* (1992) and updated by De Pinto, *et al.* (1993). Enhancements include a higher spatial resolution and linkage to a hydrodynamics model (GBHYDRO) and a sediment transport model (GBSED) of Green Bay. GBTOXe was calibrated against 1989-90 GLNPO PCB and carbon data. GBTOXe was used to run 100-year simulations of PCB fate and transport for several management scenarios, including no action.

GBTOXe is used to model total PCBs and three phases of carbon in the water column and sediments. The carbon phases considered are dissolved, biotic, and particulate detritus. The model domain consists of 1490 water column and 596 sediment segments. The water column consists of 10 layers of 149 horizontal segments. Segment volumes vary to maintain a water balance. The layers represent biologically active sediments, and deeper biologically inactive sediments. The volume of the segments in the upper 10cm of the sediment are assumed to be constant in time, while the fourth sediment layer changes in volume in response to deposition and resuspension. PCB transport mechanisms include advection, dispersion, volatilization, deposition and resuspension of sorbed phase, and pore water exchange. GBTOXe accounts for sediment bed armoring.

The GBTOXe results are published as a separate document which supplements this RI/FS/RA.

6.4.2 GBFood Model

The GBFood bioaccumulation model is a mathematical description of contaminant transfer within the food web of Green Bay zones 2 through 4. The food web is comprised of the primary energy transfer pathways from the exposure sources (sediment and water) to the fish species of interest, described in Section 4.4. These pathways include: chemical uptake across the gill surface, chemical uptake from food and chemical losses due to excretion and growth dilution. The

mathematical descriptions are generic (common to all aquatic food webs) and were updated as part of the FS.

GBFood was used in the RI/FS to estimate PCB concentrations in the food webs leading to brown trout and walleye in the Lower Fox River (from the dam at De Pere to the mouth) and Green Bay. This was accomplished by specifying values for the various physiological, bioenergetic and toxicokinetic parameters in the model and the PCB exposure levels in sediments and water. The parameter values were derived from peer reviewed studies published in the literature and/or site-specific data. The sediment and water column PCB concentrations were provided by the whole Lower Fox River Model (wLFRM) and GBTOXe model outputs. The calibrated GBFood was used to evaluate the efficacy of several remedial alternatives in reducing PCB levels in fish of the Lower Fox River and Green Bay.

The GBFood results are published as a separate document which supplements this RI/FS/RA.

6.4.3 Fox River Food (FRFood) Model

The Fox River Food (FRFood) bioaccumulation model, based on the Gobas model (1993), is a mathematical description of PCB transfer within the food web of the Lower Fox River and Green Bay (Zone 2). The model is designed to take the output of sediment and water concentrations of PCBs from wLFRM and GBTOXe to estimate concentrations in multiple trophic levels in the aquatic food web (i.e., benthic insects, phytoplankton, zooplankton, and fish). This food web model is functionally similar to, and spatially overlaps with, the food web model for Green Bay (GBFood), with the exception that the FRFood model can be run in reverse where the inputs are fish concentrations and the outputs are predicted sediment concentrations.

FRFood is based upon the algorithms originally developed for Lake Ontario PCBs (Gobas, 1993). Since then, the model has been used extensively throughout the Great Lakes, including derivation of bioaccumulation factors, bioconcentration factors, and food chain multipliers in the development of the Great Lakes Water Quality Initiative (GLWQI) criteria (EPA, 1993b; EPA, 1994). The model was first used for projecting sediment quality thresholds in the 1996 RI/FS for the Upper Fox River (GAS and SAIC, 1996), and has since been used for setting action levels at the Sheboygan River (EVS, 1998), and for predicting long-term effects on biota at the Hudson River, New York (EPA, 2000c).

The primary objectives in using the FRFood model was to 1) estimate potential risk-based remedial clean-up levels called sediment quality thresholds (SQTs), and

2) project fish tissue concentrations that would be associated with a specific remedy. To facilitate the selection of a remedy that will result in a decrease in human and ecological risks, it is necessary to establish a link between levels of PCBs toxic to human and ecological receptors, and the principle source of those PCBs, the Lower Fox River and Green Bay sediment. The FRFood model defines this link.

6.4.4 Whole Lower Fox River Model

The whole Lower Fox River Model (wLFRM) was developed from the two models developed for analysis of flow in the Lower Fox River: the Upper Fox River (UFR), which covered the river between Lake Winnebago and the De Pere dam, and the Lower Fox River model, which extended from the De Pere dam to the mouth of the river. The wLFRM retains the spatial resolution of the UFR/LFR models, but allows the simulation of the entire Lower Fox River from Lake Winnebago to the mouth of the river using a single model. The wLFRM is calibrated to data collected between 1989 and 1995. Calibration consisted of comparisons between the data and model results for total suspended solids and dissolved/particulate PCB in water, sediment bed elevation, and net sediment burial rate.

The wLFRM is used to simulate the fate and transport of solids and PCBs in the water and sediments in the Fox River. The model area is divided into 40 water column segments, 165 surficial sediment segments, and 330 subsurface sediment segments. The model predicts the movement of solids and PCBs among these various model segments. In addition, the model simulates the concentration of organic carbon in the water column. Transport mechanisms in wLFRM include advection, dispersion, volatilization, deposition, and resuspension. Deposition is a function of particle size or density with different settling rates to represent sand, silt and clay-size particles. The settling rate for clay-size particles can also be used to simulate the settling of low-density organic matter. Resuspension is based on surface water velocity and the effect of sediment bed armoring over time.

The results from the wLFRM are used as input to other modeling efforts being conducted for the Fox River/Green Bay RIFS. The wLFRM results from reaches above the De Pere dam are used as input to the FRFood model. Results from below De Pere dam to the mouth of the river are used as input to the GBFood model. Finally, the predicted solids and PCB discharges at the mouth of the river are used as inputs to the GBTOXe model.

The wLFRM results are published as a separate document which supplements this RI/FS/RA.

6.5 Section 6 Tables

Tables for Section 6 follow this page, and include:

Table 6-1 Lower Fox River - Fate and Transport Chemical Factors

Table 6-1. Lower Fox River - Fate and Transport Chemical Factors

Chemical Name	Water Solubility (mg/L)	Vapor Pressure (mm Hg)	Henry's Law Constant (atm-m ³ /mol)	Koc (ml/g)	Kow (ml/g)
Polychlorinated Biphenyls (PCBs)					
PCBs (General values)	3.10E-02	7.70E-05	1.07E-03	5.30E+05	1.10E+06
Aroclor 1016	4.20E-01	4.00E-04	2.90E-04	NA	2.40E+04
Aroclor 1221	1.50E+01	6.70E-03	3.50E-03	NA	1.23E+04
Aroclor 1232	1.45E+00	4.06E-03	NA	NA	1.58E+03
Aroclor 1242	2.40E-01	4.10E-04	5.60E-04	NA	1.29E+04
Aroclor 1248	5.40E-02	4.90E-04	3.50E-03	NA	5.62E+05
Aroclor 1254	1.20E-02	7.70E-05	2.70E-03	4.25E+04	1.07E+06
Aroclor 1260	2.70E-03	4.10E-05	7.10E-03	NA	1.38E+07
Dioxin					
2,3,7,8-TCDD	2.00E-04	1.70E-06	3.60E-03	3.30E+06	5.25E+06
Pesticides					
DDT	5.00E-03	5.50E-06	5.13E-04	2.43E+05	1.55E+06
Dieldrin	1.95E-01	1.78E-07	4.58E-07	1.70E+03	3.16E+03
SVOCs/PAHs					
Acenaphthylene	3.93E+00	2.90E-02	1.48E-03	2.50E+03	5.01E+03
Acenaphthene	3.42E+00	1.55E-03	9.20E-05	4.60E+03	1.00E+04
Anthracene	4.50E-02	1.95E-04	1.02E-03	1.40E+04	2.82E+04
Benzo(a)anthracene	5.70E-03	2.20E-08	1.16E-06	1.38E+06	3.98E+05
Benzo(a)pyrene	1.20E-03	5.60E-09	1.55E-06	5.50E+06	1.15E+06
Benzo (b)fluoranthene	1.40E-02	5.00E-07	1.19E-05	5.50E+05	1.15E+06
Benzo(ghi)perylene	7.00E-04	1.03E-10	5.34E-08	1.60E+06	3.24E+06
Benzo(k)fluoranthene	4.30E-03	5.10E-07	3.94E-05	5.50E+05	1.15E+06
Chrysene	1.80E-03	6.30E-09	1.05E-06	2.00E+05	4.07E+05
Dibenz(a,h)anthracene	5.00E-04	1.00E-10	7.33E-08	3.30E+06	6.31E+06
Fluoranthene	2.06E-01	5.00E-06	6.46E-06	3.80E+04	7.94E+04
Fluorene	1.69E+00	7.10E-04	6.42E-05	7.30E+03	1.58E+04
Indeno(1,2,3-cd)pyrene	5.30E-04	1.00E-10	6.86E-08	1.60E+06	3.16E+06
2-Methylnaphthalene	2.54E+01	NA	NA	8.50E+03	1.30E+04
Naphthalene	3.17E+01	2.30E-01	1.15E-03	1.30E+03	2.76E+03
Phenanthrene	1.00E+00	6.80E-04	1.59E-04	1.40E+04	2.88E+04
Pyrene	1.32E-01	2.50E-06	5.04E-06	3.80E+04	7.59E+04
Pentachlorophenol	1.40E+01	1.10E-04	2.75E-06	5.30E+04	1.00E+05
Bis(2-ethylhexyl) phthalate	2.85E-01	2.00E-07	3.61E-07	5.90E+03	9.50E+03
Inorganic Compounds					
Ammonia	5.30E+05	7.60E+03	3.21E-04	3.10E.00	1.00E+00
Arsenic and Compounds	NA	0.00E+00	NA	NA	NA
Barium and Compounds	NA	NA	NA	NA	NA
Cadmium and Compounds	NA	0.00E+00	NA	NA	NA
Chromium III and Compounds	NA	0.00E+00	NA	NA	NA
Chromium VI and Compounds	NA	0.00E+00	NA	NA	NA
Copper and Compounds	NA	0.00E+00	NA	NA	NA
Lead and Compounds	NA	0.00E+00	NA	NA	NA
Mercury and Compounds	3.00E-02	2.00E-03	1.10E-02	NA	NA
Nickel and Compounds	NA	0.00E+00	NA	NA	NA
Selenium and Compounds	NA	0.00E+00	NA	NA	NA
Silver and Compounds	NA	0.00E+00	NA	NA	NA
Zinc and Compounds	NA	0.00E+00	NA	NA	NA

Notes: 1) Values obtained from "Basics of Pump-and-Treat Ground-Water Remediation Technology"

EPA document EPA-600/8-90/003 or from the specific ATSDR Toxicological Profile.

2) NA - Value not available.